



Closing the global N₂O budget: nitrous oxide emissions through the agricultural nitrogen cycle

OECD/IPCC/IEA phase II development of IPCC guidelines for national greenhouse gas inventory methodology

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Abstract

In 1995 a working group was assembled at the request of OECD/IPCC/IEA to revise the methodology for N₂O from agriculture for the National Greenhouse Gas Inventories Methodology. The basics of the methodology developed to calculate annual country level nitrous oxide (N₂O) emissions from agricultural soils is presented herein. Three sources of N₂O are distinguished in the new methodology: (i) direct emissions from agricultural soils, (ii) emissions from animal production, and (iii) N₂O emissions indirectly induced by agricultural activities. The methodology is a simple approach which requires only input data that are available from FAO databases. The methodology attempts to relate N₂O emissions to the agricultural nitrogen (N) cycle and to systems into which N is transported once it leaves agricultural systems. These estimates are made with the realization that increased utilization of crop nutrients, including N, will be required to meet rapidly growing needs for food and fiber production in our immediate future. Anthropogenic N input into agricultural systems include N from synthetic fertilizer, animal wastes, increased biological N-fixation, cultivation of mineral and organic soils through enhanced organic matter mineralization, and mineralization of crop residue returned to the field. Nitrous oxide may be emitted directly to the atmosphere in agricultural fields, animal confinements or pastoral systems or be transported from agricultural systems into ground and surface waters through surface runoff. Nitrate leaching and runoff and food consumption by humans and introduction into sewage systems transport the N ultimately into surface water (rivers and oceans) where additional N₂O is produced. Ammonia and oxides of N (NO_x) are also emitted from agricultural systems and may be transported off-site and serve to fertilize other systems which leads to enhanced production of N₂O. Eventually, all N that moves through the soil system will be either terminally sequestered in buried sediments or denitrified in aquatic systems. We estimated global N₂O–N emissions for the year 1989, using midpoint emission factors from our methodology and the FAO data for 1989. Direct emissions from agricultural soils totaled 2.1 Tg N, direct emissions from animal production totaled 2.1 Tg N and indirect emissions resulting from agricultural N input into the atmosphere and aquatic systems totaled 2.1 Tg N₂O–N for an annual total of 6.3 Tg N₂O–N. The N₂O input to the atmosphere from agricultural production as a whole has apparently been previously underestimated. These new estimates suggest that the missing N₂O sources discussed in earlier IPCC reports is likely a biogenic (agricultural) one.

Introduction

The United Nations Framework Convention on Climate Change requires that all parties periodically update and publish national inventories of anthropogenic emissions by sources and removals by sinks of all greenhouse gases not controlled by the Montreal Protocol, using comparable methodologies. In response to this mandate the Intergovernmental Panel on Climate Change (IPCC), through the Office of Economic Cooperation and Development (OECD) and International Energy Agency (IEA) has been coordinating the development and updating of national inventory methodologies for various greenhouse gases. The first phase of methodology development was published in the 1995 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 1995b). In phase II a working group of 32 persons from 18 countries was assembled at the request of OECD/IPCC/IEA to revise the National Greenhouse Gas Inventories Methodology for N₂O from Agricultural Soils (IPCC, 1997). This paper presents the framework behind the phase II methodology for calculating annual, country level N₂O emissions from agricultural soils (IPCC, 1997).

During the past decade attempts to define budgets for global atmospheric N₂O suggested that the strength of known N₂O sources is underestimated or that unidentified sinks exists (IPCC, 1990, 1992; Duxbury and Mosier, 1993; Robertson, 1993). In these budgeting efforts anthropogenic N₂O emissions due to agricultural activities were considered to be relatively small (Table 1). These assessments were based upon a few reviews and interpretations that needed further examination (IPCC, 1992; Mosier, 1994; Mosier et al., 1996). Questions to these interpretations were beginning to be raised when the 1995 IPCC Guidelines for National Inventory Methodology for N₂O in Agriculture (IPCC, 1995b) was being developed (Duxbury and Mosier, 1993; Mosier, 1994; Mosier and Bouwman, 1993). Before that time N₂O emissions from agricultural systems were only considered from the aspect of direct N₂O emissions from agricultural fields (OECD/OCDE, 1991) that had been fertilized with synthetic nitrogen (N) fertilizer. The estimates used tended to underestimate total agricultural emissions (Mosier, 1994; Bouwman, 1996). For example, animal production systems have the potential for significant N₂O production (DeKlein and Logtenstijn, 1994) and needed consideration as did the remainder of the agricultural N cycle.

N applied to agricultural soils may be lost from the fields through surface erosion or leaching (Duxbury and Mosier, 1993). This leached N continues recycling in the soil–water–air system and eventually is denitrified and converted to N₂O and N₂ and released back to the atmosphere (Figure 1; Nevison et al., 1996; Oonk and Kroeze, 1998), or buried in sediments. All of these pathways and factors needed to be included in the anthropogenic agricultural soil N₂O source.

The IPCC, 1995 Guidelines (IPCC, 1995b) included N₂O emissions occurring directly from agricultural fields. The N sources in this calculation were expanded to include synthetic fertilizers, organic N from animal excreta and crop residue and the amount of biological N fixation. This basic formula equating direct N₂O emissions from agricultural soils to the N input multiplied by a conversion factor of $1.25 \pm 1.0\%$ was used in the Cole et al. (1996) Climate Change 1995 assessment of mitigation options for N₂O emissions from agriculture. The derivation of this factor is further described in Mosier et al. (1996). Values from these estimates were included in the Climate Change 1994 (IPCC, 1994) report. Cole et al. (1996) included an additional factor of 0.75% of N applications to provide some accounting for indirect N₂O emissions that eventually evolved back to the atmosphere from N leaching or runoff from agricultural fields as well as NO_x and NH₃ volatilization (Cole et al., 1996; Mosier et al., 1996) (Table 1).

The IPCC 1995 Guidelines still lacked mechanisms for estimating N-fixation and crop residue input and a quantifiable method for calculating N₂O productions following N leaching and runoff. Additionally, animal production systems were not included in the agricultural anthropogenic N₂O production guidelines. As a start in overcoming these deficiencies in National Inventory estimates, we developed a Phase II method for estimating country scale anthropogenic N₂O emissions from agricultural soils. This paper describes the essence of the methodology developed and presents calculations which suggest that an underestimation of total anthropogenic N₂O emissions from agricultural systems is responsible for the previous imbalanced global N₂O budgets.

Phase II Development of IPCC Guidelines

In 1993, the Phase I OECD/IPCC/IEA Guidelines workgroup (IPCC, 1995b) suggested that 'improving methodology for estimating N₂O emissions may evolve in a series of steps, beginning with the N

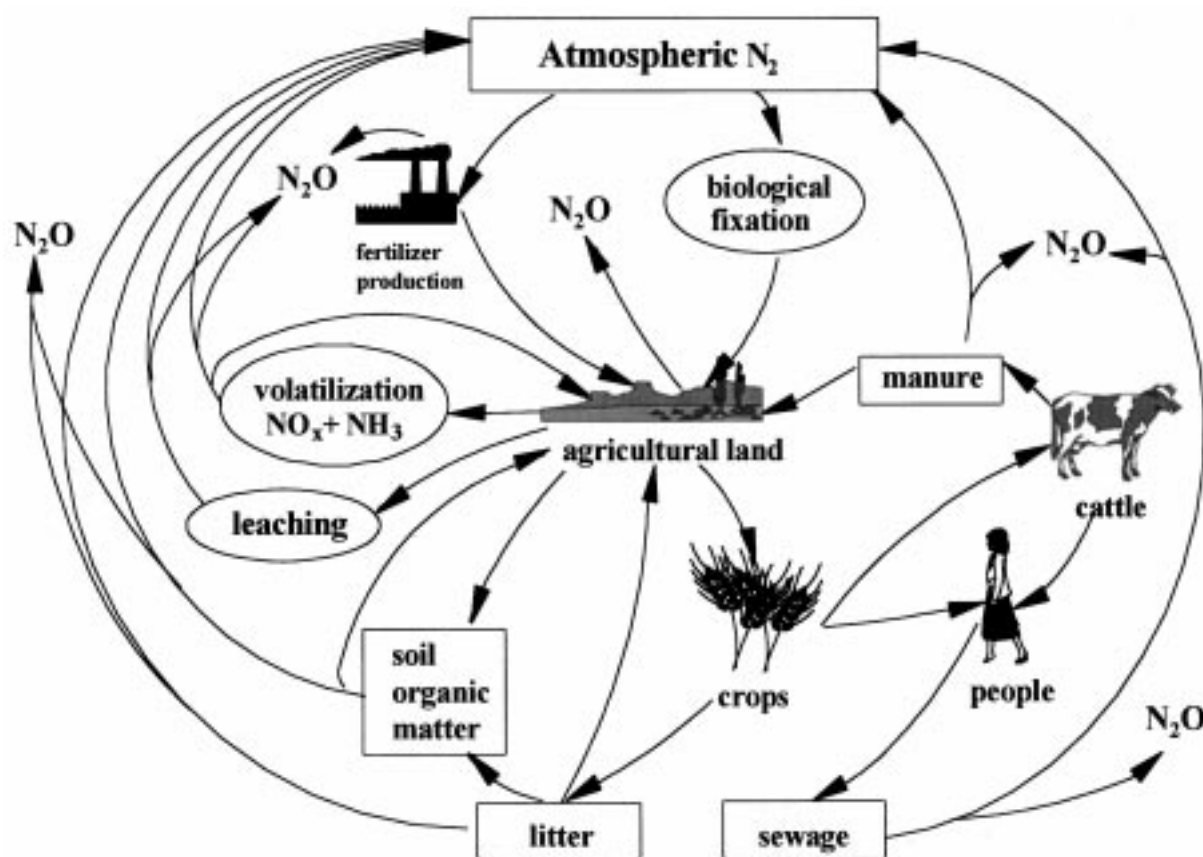


Figure 1. Depiction of the nitrogen cycle of agricultural soils and its relationship to N_2O production. (Adapted from Nevison et al., 1996; Oonk & Kroeze (1998) by permission of John Wiley & Sons, Inc.).

source based equations and ending with development of process based models which are used to develop regional and larger scale emission models'. Using this recommendation and others noted in IPCC (1995b) the Phase II IPCC/OECD/IEA work group on Agricultural Soils initiated an effort to provide a more comprehensive N_2O emission calculation methodology.

As a first step, the Phase II workgroup evaluated the IPCC Guidelines (1995b) and recommended that the emission factor relating N_2O emissions directly from the soil to fertilizer N application should be that used by Cole et al. (1996) and Mosier et al. (1997). This value, $1.25 \pm 1.0\%$ N_2O-N of fertilizer N applied (Bouwman, 1994, 1996; Cole et al., 1996; Mosier et al., 1996, 1997) was derived from field information. The range of these values cover more than 90% of the published field data summarized by Bouwman (1994, 1996). Most of the information in Bouwman's summary was derived from field studies conducted in temperate regions of the world since few annual flux

measurements have been made in tropical agricultural systems.

The second step was to develop a more comprehensive methodology which attempted to take into account the major inputs of N into agriculture, include animal sources of N_2O and start to account for the indirect production of N_2O from N released from agriculture. This methodology does not account for the impact of climate, soils, and cropping system on N_2O production, consumption and emissions, because insufficient information is available with which to define appropriate emission coefficients for each variable. We defined some general principles that were used in methodology development:

- 1) Use input data that are generally available worldwide. The data sources that best meet this requirement appear to be the agricultural yearbooks compiled by the United Nations Food and Agricultural Organization (FAO).
- 2) The general conceptual approach was adopted, in which fertilizer, animal waste, N derived from N_2 -

fixation, and crop residue N inputs to agricultural on an annual basis are tracked from their initial application to their return to the atmosphere via denitrification (Figure 1). All nitrogen flux estimates at various stages of the cycle should be consistent with the original N input and care must be taken to ensure that there is no double counting of N sources. The Phase II methodology is a first step towards such an integrated approach which assumes that all N input is reacted within one year and does not account for potential sequestration in the soil which may be released over decade to century time periods.

This new approach to estimating N₂O emissions from agricultural systems includes: (1) direct emissions of N₂O from agricultural fields (N₂ODIRECT); (2) direct emissions of N₂O in animal production systems (N₂OANIMALS) and (3) some of the indirect emission of N₂O that are derived from N that originated from agricultural systems (N₂OINDIRECT). The general equation that is the basis for the calculation development described herein follows:

$$\begin{aligned} &\text{Total N}_2\text{O-N emissions from a country} \\ &\quad (\text{kg N}_2\text{O-N y}^{-1}) \text{ are:} \\ &\text{N}_2\text{O} = \text{N}_2\text{ODIRECT} + \text{N}_2\text{OANIMALS} + \\ &\quad \text{N}_2\text{OINDIRECT} \end{aligned}$$

2. Direct N₂O emissions from agricultural soils

The IPCC Guidelines for National Greenhouse Gas Inventories only cover anthropogenic sources (i.e. the result of human activities). Anthropogenic sources of N₂O can be biogenic (e.g. enhanced N₂O production by bacteria in fertilized fields) or abiogenic (e.g. formation during burning processes). Several studies indicate that anthropogenic sources of N₂O are largely biogenic, with agriculture as a major contributor (IPCC, 1995a; Bouwman et al., 1995; Mosier et al., 1996).

Biogenic production of N₂O in the soil results primarily from the nitrification and denitrification processes. Simply defined, nitrification is the aerobic microbial oxidation of ammonium to nitrate and denitrification is the anaerobic microbial reduction of nitrate to dinitrogen gas. Nitrous oxide is a gaseous intermediate in the reaction sequences of both processes which leaks from microbial cells into the soil atmosphere (Firestone and Davidson, 1989). Major

Table 1. Global N₂O budgets: IPCC (1992), IPCC (1994) and from the N₂O methodology presented in this paper for N₂O from cultivated soils (IPCC, 1997)

	IPCC, 1992	IPCC, 1995a	IPCC, 1997
Sources	Tg N y ⁻¹		
<i>Natural^a</i>			
ocean	1.4–2.6	3(1–5)	3.0(1–5)
tropical soils			
wet forest	2.2–3.7	3(2.2–3.7)	3.0(2.2–3.7)
dry savanas	0.5–2.0	1(0.5–2.0)	1.0(0.5–2.0)
temperate soils			
forests	0.5–2.0	1(0.1–2.0)	1.0(0.1–2.0)
grasslands	?	1(0.5–2.0)	1.0(0.5–2.0)
Subtotal	4.6–8.3	9(4.3–14.7)	9.0(4.3–14.7)
<i>Anthropogenic</i>			
agricultural soils	0.03–3.0	3.5(1.8–5.3)	3.3 ^b (0.6–14.8)
biomass burning	0.2–2.1	0.5(0.2–1.0)	0.5(0.2–1.0)
industrial sources	0.8–1.8	1.3(0.7–1.8)	1.3(0.7–1.8)
cattle and feedlots	?	0.4(0.2–0.5)	2.1(0.6–3.1)
Subtotal	1.0–6.9	5.7(3.7–7.7)	7.2(2.1–19.7)
Total sources	5.6–15.2	14.7(8–22.4)	16.2(6.4–34.4)
<i>Sinks</i>			
Atmospheric increase	3–4.5	.9(3.1–4.7)	3.9(3.1–4.7)
Soils	?	?	?
Stratospheric sink	7–13	12.3(9–16)	12.3(9–16)

^aFor IPCC,1997 estimates of Natural N₂O sources we use the 'likely' values from IPCC, 1995a. The values in parenthesis in this column represent the range of estimates for each category.

^bThe 3.3 shown here is 0.9 lower than the total in Table 11, because we assume that part of the natural soil and ocean emissions estimates include part of the indirect N₂O that we calculate from emissions of NH₃ and NO_x from fertilization of agricultural soils and from nitrate leaching and runoff from these soils. The cattle and feedlot category is the animal production category from our estimates listed in Table 7.

regulators of these processes are carbon and nitrogen substrate availability, temperature, pH and soil moisture content.

In most agricultural soils biogenic formation of N₂O is enhanced by an increase in available mineral N which, in turn increases nitrification and denitrification rates. Addition of fertilizer N, therefore, directly results in extra N₂O formation (Figure 1). Most studies on N₂O emissions from agricultural soils investigate the difference in N₂O production between fertilized and unfertilized fields. Emissions from unfertilized fields are considered background emissions. However, actual background emissions from agricultural soils may be higher than historic natural emissions as a result of enhanced mineralization of soil organic matter

due to previous agricultural activities. This is particularly observed in organic soils in both cold and warm climates over the globe (Bouwman and Van der Hoek, 1991; Kroeze, 1994). Background emissions may also be lower than historic emissions due to depletion of soil organic matter (Groffman et al., 1993).

Sources of N₂O directly related to N input into agricultural soils

There are a variety of sources of N in agricultural systems that we term anthropogenic which include: (A) synthetic fertilizers, (B) animal manures (urine and feces), (C) N derived from enhanced biological N-fixation (BNF) through N₂-fixing crops (D) crop residue returned to the field after harvest and (E) human sewage sludge application. Although some part of the animal manure N, crop residue and sewage may have come from previous application of synthetic fertilizer, the reentry of this N back into the soil systems renders it again susceptible to microbial processes which produce N₂O.

A. Synthetic fertilizers and B. Animal excreta N used as fertilizer

Although synthetic fertilizers and animal manures are important sources of N₂O, their soil input is required to provide the N needed to meet global food production demands. The amount of synthetic fertilizer N applied to agricultural fields world-wide is well documented in the FAO data base (FAO Annual Yearbooks; or world wide web:

<http://www.fao.org/waicent/Agricul.htm>). Although the amount of N used as fertilizer from animal excreta is more uncertain, estimates can be made, based on animal population and agricultural practices (IPCC, 1995b; Table 5). To account for the loss of fertilizer from NH₃ volatilization and emission of nitric oxide (NO) through nitrification after fertilizer is applied to fields, an NH₃ volatilization and NO emission factor is needed. Even though climate, soil, fertilizer placement and type, and other factors influence NH₃ volatilization and NO_x emission a fixed, default emission factor of 0.1 (kg NH₃-N + NO_x-N emitted/kg N excreted) is used for synthetic fertilizers and 0.2 (kg NH₃-N + NO_x-N emitted/kg N applied) for animal waste fertilizer (Table 3) (0.2 is used for animal waste because of the potentially larger NH₃ volatilization). The amount of N from these sources available for conversion to N₂O is therefore equal to 90% of the synthetic fertil-

izer N applied and 80% of the animal waste N applied (Scheepers and Mosier, 1991).

C. Biological N fixation

Both the amount of N fixed by biological N fixation in agricultural systems and the N₂O conversion coefficient are uncertain. Biological nitrogen fixation (BNF) supplies globally some 90 to 140 Tg N yr⁻¹ to agricultural systems (Peoples et al., 1995). Although more verification on these figures is necessary, most indications are that BNF contributes more N for plant growth than the total amount of synthetic N fertilizers applied to crops each year (Danso, 1995). The Phase I IPCC Guidelines (IPCC, 1995b) mention about equal rates. On average, BNF supplies 50–60% of the N harvested in grain legumes, 55–60% of the N in nitrogen fixing trees and 70–80% of the N accumulated by pasture legumes (Danso, 1995). Cultivation of grain legumes, however, often results in net soil N depletion.

In the tropics and subtropics, the use of *Azolla* (a genus of aquatic ferns which contains an N₂-fixing cyanobacterium) is widespread. *Azolla* fixes 20–25 kg N ha⁻¹ (Kumarasinghe and Eskew, 1991) which is released upon death and decomposition. This N serves to fertilize an associated crop and eventually stimulate N₂O formation.

Galbally et al. (1992) and Bouwman and Sombroek (1990) indicate that legumes may contribute to N₂O emission in a number of ways. Atmospheric N₂ fixed by legumes can be nitrified and denitrified in the same way as fertilizer N, thus providing a source of N₂O. Additionally, symbiotically living Rhizobia in root nodules are able to denitrify and produce N₂O (O'Hara and Daniel, 1985). Galbally et al. (1992) suggest an emission rate of 4 kg N ha⁻¹ y⁻¹ for improved pastures, and Duxbury et al. (1982) suggest that legumes can increase N₂O emissions from pastures by a factor of 2 or 3. More recently Carran et al. (1995) found annual N₂O emissions ranging from 0.5 to 5 kg N₂O-N depending upon the relative fertility of the sampling location. In old and young ryegrass/clover pastures Muller (personal communication) observed N₂O emissions of 0.7 and 0.3 kg N ha⁻¹ yr⁻¹, respectively.

Because of the uncertainty in knowing the amount of N₂ fixed during N-fixation (Peoples et al., 1995) and the lack of country data on N-fixing crops, it is difficult to assign a conversion factor to N₂O emission that is related to the amount of N fixed by a crop. Total N input (FBN) is estimated by assuming that total crop biomass is about twice the mass of edible crop

(FAO, 1990b), and a certain N content of N fixing crop (FRACNRBF, Table 3). This crop production is defined in FAO crop data bases as pulses and soybeans. The N-fixation contribution does not include N_2O produced in legume pastures. This N_2O production is at least partially accounted for emissions from pastures that are being grazed. Australia and New Zealand, for example, contain large areas of pasture land that includes legumes as part of the pastoral system. Little data are available for other parts of the globe (Mosier et al., 1997).

D. Crop residue and E. Sewage sludge application

There is only limited information concerning reutilization of N from crop residues and N from sewage sludge applied to agricultural lands. Although the amount of N that recycles into agricultural fields through these mechanisms may add 25–100 Tg of $N\ yr^{-1}$ of additional N into agricultural soils (mainly from crop residues) the amount converted to N_2O is not known. To account for the N_2O in the inventory budget at this time the emission factor for fertilizers is used as default and the amount of N reentering cropped fields through crop residues is calculated from the FAO data concerning crop production.

Nitrous oxide emissions associated with crop residue decomposition are calculated here by estimating the amount of N entering soils as crop residue (FCR). The amount of nitrogen entering the crop residue pool is calculated from crop production data. Since FAO data only represent the edible portion of the crop, these must be roughly doubled to estimate total crop biomass. We assume a nitrogen percentage (FRACNCRBF and FRACNCR0; Table 3) to convert from kg dry biomass yr^{-1} to kg N yr^{-1} in crops. Some countries may have sufficient information to define the N content of crop biomass more precisely. As a default we suggest distinguishing between N-fixing crops (pulses and soybeans) and non-N-fixing crops. Some of the crop residues is removed from the field as crop (approximately 45%), and some may be burned (approximately 25% of the remaining residue in developing countries), or fed to animals. The amount of N in crop residue actually returned to a field is uncertain, as is the amount of time required for the N to mineralize. We assume here that input and impact on N_2O production occur annually. Neither the amount of root biomass remaining in the soil nor the amount of plant residue fed to animals is accounted for in this crop residue estimate.

Because no appropriate estimates of sewage sludge N used as fertilizer were found this N input is not discussed further.

Agricultural systems which may represent unusually high N_2O sources or sinks

Glasshouse farming

N-fertilizer application to glasshouse-grown crops are typically high (Postma et al., 1994). The available data are limited in scope, but three sets of studies indicate that N_2O emissions from glasshouse crops are similar to those from fields per unit of N input. Postma et al. (1994) quantified NH_3 and N_2O emissions from glasshouse cultivation of lettuce on a sandbed and found that NH_3 emissions and N_2O emitted directly or in drainage water totaled less than 1% of the N applied. Daum (personal communication) measured N_2O emissions from soilless culture cucumbers and found that N-loss rates as N_2O ranged between 0.4 and 0.9% of the N input into the culture system. Pollaris (1994) measured N_2O emission in a glasshouse cultivation of tomato and lettuce and found, respectively, 0.7 and 1.4% of the applied N emitted as N_2O . Overall, these data suggest that N_2O emissions from glasshouse agriculture do not need to be included separately in N_2O emission inventories and should be included only in the total fertilizer N consumed within each country. The importance of another factor, N_2O emission during steam disinfection of glasshouse soils, is uncertain. Postma et al. (1994) found that 2–25 kg $N_2O-N\ ha^{-1}$ were lost during 10-h following soil steaming. The extent of glasshouses to which this practice is applied is not known.

Cultivation of high organic content soils

Large N_2O emissions occur as a result of drainage and cultivation of organic soils (Histosols) due to enhanced mineralization of old, N-rich organic matter (Guthrie and Duxbury, 1978; Koops et al., 1996, 1997; Nykanen et al., 1995; Martikainen et al., 1996; Velthof et al., 1996a). The rate of N-mineralization is determined by the N-quality of the Histosol, drainage, management practices and climatic conditions. The range for enhanced emissions of N_2O due to cultivation is estimated to be 2–15 kg $N_2O-N\ ha^{-1}\ yr^{-1}$ of cultivated Histosol. Default emission values of 5 kg $N_2O-N\ ha^{-1}\ yr^{-1}$ are used for boreal and temperate regions and 10 kg $N_2O-N\ ha^{-1}\ yr^{-1}$ for tropical regions (Table 2).

Soil sink for N₂O

Aerobic soils are typically sources for N₂O, but small uptake rates have been observed in isolated instances in dry soils (Duxbury and Mosier, 1993) and in wet grass pastures (Ryden, 1981, 1983). In a seasonally burned 'cerrado' in Brazil, Nobre (1994) observed occasional small but inconsistent consumption rates and concluded that this sink was very small in these soils. Anaerobic soils have a large potential for reducing N₂O to N₂ (Erich et al., 1984), since the major product of denitrification in soils is usually N₂ rather than N₂O. However, no large, constant N₂O uptake has been reported and flooded rice fields (Parashar, 1991), for example, generally show very small emissions, depending upon the time of cropping season (Minami and Fukushima, 1984). Apparently slow rates of dissolution and transport of atmospheric N₂O in wet/or flooded soils prevents this process from being a significant regulator of atmospheric N₂O. Until additional information is available to indicate that soil uptake, in aerobic or flooded soils, is important, soil uptake of atmospheric N₂O will not be included in the N₂O budget for agricultural systems.

Methodology for estimating direct N₂O emissions from agricultural fields

Rather than repeating background information that has been published several times, we refer the reader to general discussions of conversion of different N sources to N₂O and background materials to Bouwman (1994, 1995, 1996); Mosier (1994); Mosier et al. (1996, 1997) and IPCC Guidelines (IPCC, 1995b). The following methodology for assessing direct N₂O emissions from agricultural fields includes consideration of synthetic fertilizer, N from animal waste, enhanced N₂O production due to biological N-fixation, N from crop residue mineralization and soil N mineralization due to cultivation of Histosols. In this estimate the total direct annual N₂O emission is:

$$N_2ODIRECT = [(FSN + FAW + FBN + FCR) \\ * EF1] + FOS * EF2$$

where

$$\begin{aligned} FSN &= NFERT * (1 - FRACGASF) \\ FAW &= (NEX * (1 - (FRACFUEL + FRACGRAZ + \\ &\quad FRACGASM))) \\ FBN &= 2 * CROPBF * FRACNCRBF \\ FCR &= 2 * [CROP0 * FRACNCR0 + CROPBF * \\ &\quad FRACNCRBF] * (1 - FRACR) * \\ &\quad (1 - FRACBURN) \end{aligned}$$

and

CROPBF = seed yield of pulses + soybeans in country (kg dry biomass yr⁻¹)
CROP0 = production of all other crops in country (kg dry biomass yr⁻¹)
EF1 = Emission Factor for direct soil emissions (kg N₂O-N/kg N input); Table 2
EF2 = Emission Factor for organic soil mineralization due to cultivation (kg N₂O-N ha⁻¹ yr⁻¹); Table 2
FAW = animal waste N used as fertilizer in country (kg N yr⁻¹)
FBN = N fixed by N-fixing crops in country (kg N yr⁻¹)
FCR = N in crop residues returned to soils in country (kg N yr⁻¹)
FOS = area of cultivated organic soils within a country (ha of Histosols in FAOdata base)
FRACBURN = fraction of crop residue that is burned rather than left on field; Table 3
FRACGASF = fraction of synthetic fertilizer N applied to soils that volatilizes as NH₃ and NO_x (kg NH₃-N and NO_x-N/kg of N input); Table 3
FRACFUEL = fraction of livestock N excretion contained in excrements burned for fuel (kg N/Kg N total excreted);
FRACGASM = fraction of livestock N excretion that volatilizes as NH₃ and NO_x (kg NH₃-N and NO_x-N/kg of N excreted); Table 3
FRACGRAZ = fraction of livestock N excretion contained in excrements deposited during grazing (kg N/kg N totally excreted); Appendix 1
FRACNCRBF = fraction of N in N-fixing crop (kg N/kg of dry biomass); Table 3
FRACNCR0 = fraction of N in non-N-fixing crop (kg N/kg of dry biomass); Table 3
FRACR = fraction of crop residue that is removed from the field as crop (kg N/kg crop-N); Table 3
FSN = synthetic N applied in country (kg N yr⁻¹)
N₂ODIRECT = direct N₂O emissions from agricultural soils in country (kg N yr⁻¹)
NEX = amount of N excreted by the livestock within a country (kg N yr⁻¹); Table 5
NFERT = synthetic fertilizer use in country (kg N yr⁻¹)

The input data needed for this methodology include synthetic fertilizer use (NFERT), manure-N used as fertilizer (FAW), edible crop production of N-fixing crops (CROPBF) and non-N-fixing crops (CROP0), and area of cultivated organic soils (Histosols) in the country. The data for synthetic fertilizer

Table 2. Summary of default emission factors for agricultural emissions of N₂O

EF1 = 0.0125 (0.0025 – 0.0225) kg N ₂ O–N/kg N input
EF2 = 5 temperate and 10 tropical (2–15) (kg N ha ⁻¹ yr ⁻¹)
EF3: see Table 6
EF4 = 0.01 (0.002 – 0.02) kg N ₂ O–N/kg NH ₃ –N and NO _x –N emitted
EF5 = 0.025 (0.002 – 0.12) kg N ₂ O–N/kg N leaching/runoff
EF6 = 0.01 (0.002 – 0.12) kg N ₂ O–N/kg sewage-N produced

use are available on a country basis in the FAO data base (e.g. FAO, 1990a) and the amount of N in animal waste applied to agricultural fields (FAW) is calculated from the number and type of animals within a country (FAO data base; IPCC, 1995b) and an in-country estimate of the percentage of N excreted by farm animals that is collected during confinement and reapplied to the field (Table 5 and Appendix 1). Both synthetic fertilizer and manure used as fertilizer need to be corrected for the amount of NH₃ volatilized and NO_x emitted (10 and 20% of N applied or excreted, respectively) after the material is placed in or on the soil so that the same N atom is not counted again. The FAW data also need to be carefully evaluated for each country to be sure that animal waste used to fertilize crops and animal waste deposited on pastures while animals are grazing are not double counted. Crop production data for pulses and soybeans and non-N-fixing crops are listed in the FAO crop data base (FAO, 1990b).

3. Direct N₂O emissions in animal production

N₂O Sources within animal production systems

Earlier IPCC estimates (IPCC, 1995a) of N₂O emission from agriculture and other sources (IPCC, 1990, 1992) did not include N₂O emission from animal production. Recent studies (e.g. Bouwman, 1995; Jarvis and Pain, 1994; Flessa et al., 1996; Mosier et al., 1996, 97) indicate that emissions from animal wastes can be significant. There are three potential sources in animal production, i.e. (A) animals themselves, (B) wastes from confined animals and (C) dung and urine deposited on the soil by grazing animals. Emissions induced by use of manure N as fertilizer (with the exception of grazing animals) are considered direct N₂O emissions from agricultural fields and are included in the previous section of this paper.

A. N₂O from animals

Animals themselves may be very small sources of N₂O. Animal fodders contain 10 to 40 g of N/ kg dry matter. The greater part of this N is organically bound, but as total N content increases so does the nitrate (NO₃⁻) content, generally. Nitrate contents in fodders generally range from 1–10 g N/ kg dry matter (Spoelstra, 1985). Upon passage through the digestive track of the animal, nitrate is reduced via dissimilatory nitrate reduction to NH₃/NH₄⁺. The nitrate reduction reaction may release small amounts of N₂O in the gut (Kaspar and Tiedje, 1981), which may escape to the atmosphere during rumination. Though this possible route of N₂O formation has been known for over 10 years, quantitative data in terms of N₂O release are still lacking to-date. The total amount of N₂O released by cattle is probably very small, because the gut is highly anoxic and this will favor the formation of NH₃/NH₄⁺ (Tiedje, 1988). Direct losses from animals themselves are likely to be much less than 10 g N₂O–N/ kg N excreted or taken up by the animal. Therefore, it is not included in the emission estimate.

B. N₂O emissions from animal waste management systems

The proportion of total N intake that is excreted and its partition between urine and feces is dependent on the type of animal, the intake of dry matter, and the N concentration of the diet (Whitehead, 1970). The retention of N in animal products (i.e. milk, meat, wool and eggs) generally ranges from about 5 to 20% of the total N intake. The remainder is excreted via dung and urine. For sheep and cattle, fecal excretion is usually about 8 g/kg of dry matter consumed, regardless of the N content of the feed (Haynes and Williams, 1993). The remainder of the N is excreted in the urine and as the N content of the diet increases, so does the proportion of N in the urine. In intensive animal production systems, where animal intake of N is high, more than half of the N is excreted as urine.

Production of N₂O during storage and treatment of animal wastes can occur via combined nitrification-denitrification of ammoniacal N contained in the wastes. The amount released depends on the system and duration of waste management. As fresh dung and slurry is highly anoxic and well-buffered with near neutral pH, one would expect N₂O production to increase with increasing aeration. Aeration initiates the nitrification-denitrification reactions, and hence makes release of N₂O possible. Unfortunately, there is not enough quantitative data to derive a relationship be-

Table 3. Summary of default values for parameters

FRACBURN	= 0.25 in developing countries; 0 in developed countries (kg N/kg crop-N)
FRACFUEL	= justified in-country estimate, or 0.0 kg N/kg N excreted
FRACGASF	= 0.1 kg NH ₃ -N + NO _x -N/kg of synthetic fertilizer N applied
FRACGASM	= 0.2 kg NH ₃ -N + NO _x -N/kg of N excreted by livestock
FRACGRAZ	= Tables 5,6 and Appendix 1
FRACLEACH	= 0.3 kg N/kg N of fertilizer or manure
FRACNCRBF	= 0.03 kg N/kg of dry biomass
FRACNCR0	= 0.015 kg N/kg of dry biomass
FRACNPR	= 0.16 kg N/kg of protein
FRACR	= 0.45 kg N/kg crop-N

tween the degree of aeration and N₂O emission from slurry during storage and treatment. There is a wide range in estimated losses, when expressed in g N₂O N per kg N in the waste. Losses from animal waste during storage range from < 0.1 g N for slurries to > 150 g N per kg N for pig waste in deep-litter stables (Groenestein et al., 1993; Kroeze, 1994; Sibbesen and Lind, 1993). Hence, different emission factors are needed for different animal waste management systems (Table 6). We use emission factors of 1, 20, and 5 g N₂O-N kg⁻¹ of N excreted for anaerobic lagoons and liquid systems, solid storage and drylot and for all other animal waste management systems, respectively. These emission factors should be revised as new data become available.

C. N₂O from animal grazing

A brief summary of estimates of N₂O emissions derived from dung and urine deposits of grazing animals is compiled in Table 4. The N₂O emission is expressed as per cent of the N in urine and/or dung. Two types of studies may be distinguished. The first type focuses on emissions from a well-defined urine and/or dung patch. A control treatment is generally included, to facilitate the calculation of urine and dung derived emissions. The grazed grassland is the focus in the second type of studies. Grazing derived emissions can be obtained properly when a non-grazing treatment is included. For the purpose of this compilation we consider that grazing derived emissions are similar to 'dung and urine derived' emissions. This may not be completely true, because grazing animals have also other effects than deposition of dung and urine. For example, compaction of the soil by trampling may increase N₂O emission from soil (Oenema et al., 1997).

The duration of the studies ranged from 1 week up to 2 years. Though the bulk of the N₂O will be lost shortly after deposition in the field, significant amounts may still be released from the urine and dung even after a couple of weeks after deposition. Hence, short-term studies may under-estimate the total N₂O losses from animal excrements (Van Cleemput et al., 1994; Velthof et al., 1996a). Grazing derived emissions range from 2 to 98 g N₂O-N/ kg of N excreted (Table 4). The lower estimates are from well-drained unfertilized grassland soils in New Zealand (Carran et al., 1995). Large grazing derived emissions, induced by livestock N excretion, were obtained on drained peat soils in the Netherlands. These intensively managed grassland on peat soils have also a large background emission and also a large fertilizer derived emission (Velthof and Oenema, 1995; Velthof et al., 1996a).

Nearly all data pertain to temperate areas, with intensively managed grassland. The N contents of dung and especially urine are higher from these intensively managed grassland than from the less intensively managed (sub)tropical grasslands. The fraction of easily hydrolyzable N, i.e. urea and uric acid, is much smaller in dung and urine from animals fed with a low N content ration than from animals fed with a high N content ration. This difference will probably result in a different emissions factor. Unfortunately, data are lacking to sustain this hypothesis. Differences in climate, i.e. rainfall and temperature patterns, may also have a significant effect.

Nitrogen losses as N₂O are probably lower in arid and semiarid regions. Mosier and Parton (1985) found that during the course of a year in a semiarid grassland per kg of urea-N from simulated urine patches, only 6 g N₂O-N was emitted as N₂O. They did find in

Table 4. Emission of N₂O from animal dung and urine deposited in grassland; a compilation of published and unpublished data. The emitted amount of N₂O is expressed in % of the amount of N excreted by the grazing animal (see also Oenema et al., 1997)

Country	Soil type	Treatment	Period	N ₂ O Emission	Reference
United Kingdom	clay loam	urine	4 weeks	1-5	Monaghan and Barraclough (1993)
New Zealand	silt loam	urine	6 weeks	<0.5	Sherlock and Goh (1983)
Germany	loess	urine	11 weeks	3.8	Flessa et al. (1996)
Germany	Loess	dung	11 weeks	0.5	Flessa et al. (1996)
The Netherlands	clay	urine	4 weeks	0.5	Velthof and Oenema (1994)
United Kingdom	clay loam	grazing	1 week	1.8	Velthof et al. (1996b)
The Netherlands	sand	grazing	32 weeks	1.0	Velthof and Oenema (1995)
The Netherlands	peat I	grazing	32 weeks	1.5	Velthof and Oenema (1995)
The Netherlands	peat II	grazing	32 weeks	7.7	Velthof and Oenema (1995)
Germany	loam	urine/dung	1 year	0.4-1.3	Poggeman et al. (1995)
The Netherlands	sand	grazing	2 years	1.5	Velthof et al. (1996a)
The Netherlands	clay	grazing	2 years	3.3	Velthof et al. (1996a)
The Netherlands	peat I	grazing	2 years	2.3	Velthof et al. (1996a)
The Netherlands	peat II	grazing	2 years	9.8	Velthof et al. (1996a)
New Zealand	silt loam	grazing	1 year	0.2-1.0	Carran et al. (1995)

later studies that N₂O emissions remained detectably higher 10 years after the urea had been applied to the semiarid shortgrass prairie (Mosier et al., 1991).

An overall reasonable average emission factor for animal waste excreted in pastures is 20 g N₂O–N/kg of N excreted. Note that this emission factor is almost twice as large as the emission factor for animal wastes spread on agricultural fields following manure storage. This difference is thought to be related to the high N content in the dung and urine patches.

Methodology for estimating N₂O from animal production

A simple methodology which takes into account the type and number of animals, the N excretion by the animals, and the Animal Waste Management System (AWMS), which includes free-range grazing is described below. Specific emission factors, expressed in per cent of N excreted are assigned to the animals and to the AWMS, representing direct emissions from the AWMS. Hence, the total emission from animal production in a country follows from:

$$N_2OANIMALS = \Sigma(N_2OAWMS)$$

where

$$N_2OAWMS = [N(T=1) * NEX(T=1) * AWMS(T=1) * EF3(AWMS)] + \dots + [N(T=TMAX) * NEX(T=TMAX) * AWMS(T=TMAX) * EF3(AWMS)]$$

and:

AWMS(T) = fraction of NEX(T) that is managed in one of the different distinguished animal waste management systems for animals of type T in the country, Appendix 1

EF3(AWMS) = N₂O emission factor for an AWMS (kg N₂O–N/kg of NEX in AWMS), Table 6

N₂OANIMALS = N₂O emissions from animal production and pastures (kg N yr⁻¹) from a country

N₂OAWMS = N₂O emissions from Animal Waste Management Systems in the country (kg N yr⁻¹)

N(T) = number of animals of type T in the country

NEX(T) = N excretion of animals of type T in the country, Table 5

T = type of distinguished animals

TMAX = last type of animal distinguished in the country

Nitrogen excretion

General statistics about animal numbers are provided by FAO in their Production Yearbooks (FAO, 1956–

Table 5. Default values for N excretion per head of animal per region (kg N animal⁻¹ yr⁻¹)

Region	Type of animals					
	Non-dairy cattle	Dairy cattle	Poultry	Sheep	Swine	Other animals
North America	70	100	0.6	16	20	25
Western Europe	70	100	0.6	20	20	25
Eastern Europe	50	70	0.6	16	20	25
Oceania	60	80	0.6	20	16	25
Latin America	40	70	0.6	12	16	40
Africa	40	60	0.6	12	16	40
Near East & Mediterranean	50	70	0.6	12	16	40
Asia & Far East	40	60	0.6	12	16	40

1994) and detailed information is available for many countries. Default values for N excretion for each animal type and region are provided in Table 5, which was compiled on the basis of data provided by ECE-TOC (1994), and references therein, Vetter et al. (1988), Steffens and Vetter (1990). There are still uncertainties in the values listed in Table 5. Estimates for cattle and swine may be too high. Hence, these estimates (default values) need further attention. Countries may choose to use N excretion data from the Ammonia Expert Panel of the UN-ECE task force on emission inventories.

Animal waste management systems

The types of Animal Waste Management Systems (AWMS) distinguished by Safley et al. (1992) and their compilations for a large number of countries are proposed for this methodology (Appendix 1). Descriptions of these management systems can be found in Vol. 3 of the IPCC Guidelines, in Table 4–8 and Table 4–6 (IPCC, 1995b). The AWMS is an important regulating factor in N₂O emission from animal wastes during storage and treatment. The data provided per country in Safley et al. (1992) could be used for estimating N₂O emissions from animal wastes. Significant differences in emission factors are expected between some of the AWMS.

The class ‘used for fuel’ is not included here as a source of N₂O, because this possible source of N₂O is considered an energy-related emission. Nevertheless, countries should estimate the amount of manure N that is used as fuel, because that amount is not applied to soils (see Appendix 1).

Default emission factors (EF3) for the different AWMS are shown in Table 6. These factors were

Table 6. Tentative default values for N₂O emission factors from animal waste per Animal Waste Management System, in g N₂O–N per kg N excreted

Animal Waste Management System ^a	Emission factor EF3
Anaerobic lagoons	1 (0 - 2)*
Liquid Systems	1 (0 - 1)
Daily spread	0.0 (no range) ^b
Solid storage & drylot	20 (5 - 30)
Pasture range & paddock (grazing)	20 (5 - 30)
Used as fuel	not included
Other systems	5 (no range)

*indicates range

^a See Appendix 1.

^b Assumes no storage of manure; emissions after use as fertilizer are included in direct soil emissions.

derived on the basis of a very limited amount of information. Uniform factors for all over the world are proposed. This may be incorrect, as temperature and moisture may have positive effects on the size of the processes and, hence, on losses. However, animal production systems in warmer regions, are low-intensity systems and generally have less easily hydrolyzable N in the excretions, therefore, less N₂O production. These counteracting effects suggest that a uniform factor for all regions would seem appropriate.

4. Indirect N₂O emissions from N used in agriculture

Overview of sources for indirect N₂O production

The pathways for synthetic fertilizer and manure nitrogen input that give rise to indirect emissions are considered to be:

- Volatilization and subsequent atmospheric deposition of NH₃ and NO_x
- Nitrogen leaching and runoff
- Human consumption of crops followed by municipal sewage treatment
- Formation of N₂O in the atmosphere from NH₃
- Food processing

Of these, methodologies for estimating N₂O emissions from A–C are proposed. At present, insufficient information is available to estimate emissions from D and E.

A. Atmospheric deposition of NO_x and NH₃

Atmospheric deposition of nitrogen compounds such as nitrogen oxides (NO_x) and ammonium (from NH₃)

fertilize soils and surface waters and as such enhance biogenic N_2O formation. Indeed, Brumme and Beese (1992) showed that after two decades of atmospheric deposition of acidifying compounds (ammonium and sulphuric acids), N_2O emissions from German forest soils were enhanced by up to a factor of 5. Reported rates of N_2O emissions are between 0.2 and 1.6% of the amount of N deposited onto soils (Bowden et al., 1991; Brumme and Beese, 1992; Kasimir, personal communication). This is within the range of emission factors suggested for synthetic fertilizers. We therefore calculate N_2O -N emissions as 1% of NO_x -N and NH_3 -N emission in a country (EF4, Table 2).

Animal manure (dung + urine) is one of the most important sources of atmospheric NH_3 . According to Van der Hoek (1994), up to 50% of the mineral N in animal manure (i.e. about 25% of total N) may be lost shortly as NH_3 after application to soil. He also shows that this percentage depends considerably on the application technique used. Schimel et al. (1986) assumed that, as a minimum estimate, 20% of manure nitrogen applied to soils is volatilized as NH_3 soon after application. Although climate and fertilizer type (e.g. Urea or ammonium sulfate) may influence ammonia volatilization, we use the following default for NH_3 and NO_x volatilization: 0.1 kg N/ kg synthetic fertilizer N applied to soils and 0.2 kg N/ kg of N excreted by livestock (FRACGASF and FRACGASM, Table 3). These N_2O -N emissions are to be calculated from a country's NO_x and NH_3 emissions and N transported in leaching and runoff, so that all N_2O formed as a result of NO_x and NH_3 emissions and leaching and runoff in country Z are assigned to country Z, even if the actual N_2O formation takes place in another country. This implies that NO_x and NH_3 and N from leaching and runoff imported into a country is not included in the calculations.

B. Leaching and runoff

A considerable amount of fertilizer N is lost from agricultural soils through leaching and runoff. The leached/runoff N enters groundwater, riparian areas and wetlands, rivers and eventually the coastal ocean. In many world areas it is one of the most important inputs of nitrogen to those systems. A WHO/UNEP report (1989) showed that over 10% of European rivers had a nitrate content ranging from 9 to 25 mg nitrate-N/L; in the US 10 mg nitrate-N/L is considered unfit for human consumption. Mineral N in ground water and surface waters enhances biogenic

production of N_2O as the N undergoes nitrification and denitrification.

The amount of nitrogen lost to leaching and surface runoff may differ considerably between watersheds (Kroeze and Seitzinger, this volume). We therefore estimate that the fraction of the fertilizer and manure N lost to leaching and surface runoff (FRACLEACH) may range from 0.1–0.8. For this purpose total N excretion (NEX) is used in order to include manure produced during grazing:

$$\text{NLEACH} = [\text{NFERT} + \text{NEX}] * \text{FRACLEACH}$$

The sum of the emission of N_2O due to NLEACH in: 1) groundwater and surface drainage (EF5-g), 2) rivers (EF5-r), and 3) coastal marine areas (EF5-e) is calculated to obtain the N_2O emission factor (EF5) for NLEACH. Although not specified, the total amount of N eventually denitrified remains the same but some is denitrified in riparian area and wetlands before the N reaches the ocean. In future assessment methodologies, a separate emission factor should be used for each of these three environments.

Background information for developing emission factor (EF5) for NLEACH.

Groundwater and surface drainage. Supersaturated concentrations of N_2O in groundwater and in surface water draining agricultural lands may occur due to leaching of N_2O from the soil towards drainage and groundwater, or production during nitrification and/or denitrification of fertilizer N in the groundwater or drainage ditches. Mineral N in groundwater or drainage water is primarily in the form of NO_3 -N. While the range of N_2O concentrations reported is large, there is some relationship between the concentration of N_2O -N and NO_3 -N in groundwater and agricultural drainage water. The ratio of N_2O -N to NO_3 -N concentration in groundwater and agricultural drainage water at over 25 locations in urban, agricultural and woodland areas in Japan, Israel and the United States ranged from 0.0001 to 0.06 (Dowdell et al., 1979; Minami and Fukuski, 1984; Ronen et al., 1988; Minami and Oshawa, 1990; Ueda et al., 1991; Ueda et al., 1993). The ratio of N_2O -N to NO_3 -N in agricultural drainage ditches and groundwater under agricultural fields ranged from approximately 0.0003 to 0.06. The ratio of N_2O -N to NO_3 -N in agricultural drainage ditches were generally lower (0.003 or less) than ratios in agricultural groundwater. Rapid loss of N_2O to the atmosphere may account for the generally lower ratios in drainage ditch water. The ra-

tio of $\text{N}_2\text{O-N}$ to $\text{NO}_3\text{-N}$ in agricultural groundwater was generally between 0.003 and 0.06, with values between 0.007 and 0.02 common. Assuming that all NLEACH is in the form of NO_3 , we recommend a default emission factor of 0.015 (EF5-g) for N_2O from NLEACH in groundwater and drainage ditches, with a range of 0.003 to 0.06. The amount of N_2O emitted from groundwater (by upward diffusion or following entry of groundwater into surface water through rivers, irrigation, and drinking water) and agricultural drainage water is then estimated as:

$$\text{NLEACH} * \text{EF5-g}$$

where $\text{EF5-g} = 0.015 \text{ kg } \text{N}_2\text{O-N} / \text{kg NLEACH}$, assuming that all N_2O produced in a particular year is emitted during that year.

Rivers. Once NLEACH from groundwater and surface water enters rivers, additional N_2O is produced associated with nitrification and denitrification of NLEACH (Seitzinger and Kroeze, in prep.).

Nitrification: N_2O can be produced during nitrification of NLEACH in rivers. While most of the NLEACH enters rivers as nitrate, algae and aquatic plants can assimilate the nitrate into organic matter, which is released as ammonia, following decomposition of that organic matter. Ammonia in rivers is rapidly nitrified (Lipschultz et al., 1986). The NLEACH entering rivers nitrifies on average 0.5 – 3 times during river transport. We assume for our default methodology that all NLEACH entering rivers is nitrified once during river transport. The N_2O yield (moles $\text{N}_2\text{O-N}$ per mol of $\text{NO}_3\text{-N}$) during nitrification is generally between 0.002 and 0.003 at atmospheric oxygen levels (0.2 atm partial pressure) (Watson, 1965; Watson and Remsen, 1969; Goreau et al., 1980), enhanced yields of N_2O are found at reduced O_2 concentrations (Goreau et al., 1980). We use an N_2O yield of 0.005 for nitrification.

Denitrification: During river transport a considerable amount of nitrogen is lost via denitrification. A wide range of denitrification rates has been measured in riverine sediments; rates are generally lowest in unpolluted streams (Duff et al., 1984) with highest rates in polluted rivers/streams (Robinson et al., 1979; Cooper and Cooke, 1984; Seitzinger, 1988, 1990; Christensen and Sorensen, 1988; Christensen et al., 1989). Factors likely to affect the fraction of N removed by denitrification include length and depth of the river, flow rate, water residence time, oxygen content, organic content of sediments, and season. In

a number of rivers denitrification removed 50% of the N inputs, even over short sections (Kaushik and Robinson, 1976; Hill, 1979, 1981, 1983; van Kessel, 1977; Swank and Caskey, 1982). Here, we assume that denitrification removes 50% of NLEACH inputs to rivers. N_2O associated with denitrification (Jorgensen et al., 1984) is released from river sediments. The ratio of $\text{N}_2\text{O:N}_2$ emitted from river sediments is generally within the range 0.001 – 0.005, although in heavily polluted sediments yields up to 6% have been observed (Seitzinger, 1988). A constant ratio of 0.005 for $\text{N}_2\text{O-N}$ emission to denitrification ($\text{N}_2\text{-N}$ production) in rivers is suggested.

In summary, the emission factor for NLEACH in rivers due to nitrification and denitrification [EF5-r] is thus equal to $0.005 * \text{NLEACH}$ [for nitrification] plus $0.005 * (\text{NLEACH}/2)$ [for denitrification], or $0.0075 * \text{NLEACH}$. Therefore, $\text{N}_2\text{O-N}$ produced from NLEACH during river transport = $\text{NLEACH} * (\text{EF5-r})$, where $\text{EF5-r} = 0.0075$.

Estuaries. Rivers are the major conduit for N transport to the coastal ocean (estuaries). As discussed above, half of NLEACH is assumed to be removed by denitrification in rivers in the form of N_2 and N_2O . The remaining 50% of NLEACH is discharged by rivers to estuaries. N inputs to estuaries can undergo nitrification and denitrification, with associated N_2O production.

Nitrification: Nitrification rates in estuaries generally range from $0\text{--}22 \mu\text{mol l}^{-1} \text{ d}^{-1}$ (Berounsky and Nixon, 1993). Estuarine nitrification rates are affected by a number of factors such as ammonia concentrations, temperature (Berounsky and Nixon, 1985, 1993), oxygen (Helder and DeVries, 1983), suspended particulate matter (Helder and DeVries, 1983; Owens, 1986), and light (Horriggan and Springer, 1990). However, no predictive factor has been developed to estimate pelagic nitrification rates across a range of estuaries. In Narragansett Bay (USA), approximately half of the river inputs of inorganic N to the Bay were nitrified in the bay (Berounsky and Nixon, 1993; Seitzinger and Kroeze, in prep.). Herein, we assume that half of the rivers inputs of NLEACH are nitrified again in estuaries, and that the ratio of $\text{N}_2\text{O-N}$ to $\text{NO}_3\text{-N}$ produced is 0.005, as discussed above for rivers.

Denitrification: A relatively good relationship has been found between denitrification and inorganic N inputs to estuaries from rivers (Kemp et al., 1990; Jenkins and Kemp, 1984; Jensen et al., 1984, 1988;

Smith et al., 1985) which is equivalent to a relatively constant percentage (50%) of inorganic N inputs to a variety of estuaries (Seitzinger, 1988). Those estuaries vary in a number of characteristics including N loading rates (25 to $516 \times 10^{-6} \text{ mol N m}^{-2} \text{ h}^{-1}$), extent of intertidal area ($< 1\%$ to 50%), and latitude (subtropical to subarctic). We use 50% of the NLEACH that is carried to estuaries by rivers is denitrified, and the ratio of $\text{N}_2\text{O-N}$ to denitrification ($\text{N}_2\text{-N}$) emitted is 0.005, as discussed above for rivers. NLEACH that enters estuaries but is not denitrified, is either buried in the sediments as organic N or exported to the continental shelf region where additional N_2O can be produced. Nitrous oxide production associated with this fraction of NLEACH is not accounted for in this methodology.

In summary, the Phase II methodology assumes the following: 1) half of the NLEACH is transported to estuaries by rivers, 2) half of the NLEACH in estuaries is nitrified again in the estuary with a ratio of $\text{N}_2\text{O-N}$ to $\text{NO}_3\text{-N}$ of 0.005, and 3) half of the NLEACH in estuaries is denitrified in the estuary with a $\text{N}_2\text{O-N}$ to $\text{N}_2\text{-N}$ ratio of 0.005. Therefore, $\text{N}_2\text{O-N}$ produced from NLEACH in estuaries = $\text{NLEACH} \times (\text{EF5-e})$ where $\text{EF5-e} = 0.0025$.

The combined emission factor [EF5] for N_2O due to NLEACH in: 1) groundwater and surface drainage ($\text{EF5-g} = 0.015 \text{ kg N}_2\text{O-N per kg NLEACH}$), 2) rivers ($\text{EF5-r} = 0.0075 \text{ kg N}_2\text{O-N per kg NLEACH}$), and 3) coastal marine areas ($\text{EF5-e} = 0.0025 \text{ kg N}_2\text{O-N per kg NLEACH}$) is 0.025 (EF5). Therefore:

$$\text{NLEACH} = [\text{NFERT} + \text{NEX}] \times \text{FRACLEACH}$$

$$\text{N}_2\text{O(L)} = \text{NLEACH} \times \text{EF5}$$

where the default values are $\text{FRACLEACH} = 0.3 \text{ kg N/kg N input to soils}$ and $\text{EF5} = 0.025 \text{ kg N}_2\text{O-N/kg NLEACH}$.

C. Human consumption followed by municipal sewage treatment

Consumption of foodstuffs by humans results in the production of sewage. Sewage can be disposed of directly on land (night-soil or spray irrigation) or discharged into a water source (e.g. rivers and estuaries). Before disposal on land or into water, it also can be processed in septic systems or wastewater treatment facilities. During all of these stages, N_2O can be produced during nitrification and denitrification of sewage nitrogen. In the 1997 Guidelines (IPCC, 1997) N_2O emissions from sewage will be reported as waste emissions rather than under agricultural sources.

Sewage N (NSEWAGE) production can be estimated from FAO per capita protein consumption data (PROTEIN) and human population counts (NRPEOPLE), assuming that N constitutes about 16% by weight of protein (FRACNPR, Table 3): Sewage N production can be estimated from FAO per capita protein consumption data (PROTEIN) and human population counts (NRPEOPLE). Protein consumption may vary by a factor of 2 between countries, e.g., Americans and Indians consume 110 and 55 g protein/person/day, respectively.

$$\text{NSEWAGE} = \text{PROTEIN} \times \text{FRACNPR} \times \text{NRPEOPLE}$$

Nitrous oxide emissions resulting from sewage nitrogen are estimated following: 1) land disposal or wastewater treatment of sewage, and 2) input of sewage N to rivers and estuaries.

Disposal or wastewater treatment of sewage. No studies were found quantifying nitrous oxide emissions from land disposal of sewage, although supersaturated concentrations of N_2O in groundwater under cultivated land irrigated with sewage effluent have been reported (Ronen et al., 1988). A few studies have documented N_2O emission associated with wastewater treatment operations (e.g. Benckiser et al., 1996; Czepiel et al., 1995; Debruyne et al., 1994; Hemond and Duran, 1989; Hanaki et al., 1992; Hong et al., 1993; Velthof and Oenema, 1993). All studies reported low rates of N_2O emission. For example, nitrous oxide emissions from a secondary treatment wastewater facility in New Hampshire (USA) were approximately $0.0006 \text{ g N}_2\text{O-N/g sewage N}$, assuming $3.2 \text{ kg sewage N are produced person}^{-1} \text{ yr}^{-1}$ (Czepiel et al., 1995). Velthof and Oenema (1993) found N_2O losses of 22 g d^{-1} in a vented closed waste water treatment facility that had a daily input of 900 kg N , suggesting that N_2O losses were $0.00005 \text{ g/g N entering the system}$. Additional N_2O released to the atmosphere following discharge of supersaturated effluent to the environment is also low ($0.0007 \text{ g N}_2\text{O-N/g sewage N}$) (Hemond and Duran, 1989).

For the our estimates N_2O associated with sewage treatment and land disposal is assumed to be negligible. This is based on the low emission rates of N_2O reported for operating wastewater treatment facilities (Hemond and Duran, 1989; Czepiel et al., 1995; Velthof and Oenema, unpubl. results), and the lack of information on N_2O production from land disposal of human sewage. This assumption should be reviewed in the future, as new data become available.

Rivers and estuaries. N_2O is produced in rivers and estuaries following nitrification and denitrification of sewage N inputs (Seitzinger and Kroeze, in prep.). The sewage nitrogen can be discharged directly to aquatic environments (e.g. rivers, estuaries) or enter aquatic environments following leaching from terrestrially disposed sewage. Here, it is assumed that minimal removal of sewage nitrogen occurs during land disposal or sewage treatment, and that all sewage nitrogen enters rivers and/or estuaries. This latter assumption should be reviewed in the future, as more data become available.

Nitrous oxide emissions in rivers and estuaries due to nitrification and denitrification of sewage N are estimated using the same assumptions used for mineral N leached to rivers and estuaries (Section B). These assumptions result in emission coefficients of $EF6-r = 0.0075 \text{ g } N_2O-N/g \text{ NSEWAGE (rivers)}$ and $EF6-e = 0.0025 \text{ g } N_2O-N/g \text{ NSEWAGE (estuaries)}$. The sum of N_2O emissions in rivers ($0.0075 * \text{NSEWAGE}$) and estuaries ($0.0025 * \text{NSEWAGE}$) associated with nitrification and denitrification of NSEWAGE is calculated as:

$$N_2O(S) = \text{NSEWAGE} * EF6$$

where $EF6 = 0.01 \text{ g } N_2O-N/ \text{ g NSEWAGE}$

D. Formation of N_2O in the atmosphere from NH_3

Dentener and Crutzen (1994) proposed that oxidation of NH_3 and subsequent reaction of the intermediate NH_2 radical with NO_2 could lead to a production of $0.6 + 0.3 \text{ Tg } N_2O-N \text{ yr}^{-1}$. They parameterized natural NH_3 emissions from vegetation using a highly uncertain NH_3 canopy compensation point (the atmospheric concentration above which plants assimilate and below which they emit NH_3). Without considering this compensation point, N_2O production was reduced by 55%. Other sources of NH_3 in the tropics include animal waste decomposition (both from wild and domestic animals), fertilizer application and biomass burning emissions. About half of the atmospheric N_2O production may be associated with agricultural nitrogen, amounting to about $0.25 \text{ Tg } N_2O-N \text{ yr}^{-1}$. Due to the high uncertainty of this estimate (ca. 100%), we have not included this potentially important source in our agricultural N_2O emissions inventory. More measurements on the co-occurrence of high NH_3 , NO_2 and OH concentrations in the tropics are needed to provide more insight in the photochemical production of N_2O .

E. Food processing operations

Some food processing operations are sources of N_2O . A fraction of the edible crop harvest is not consumed by people and enters the waste stream, for instance when it is landfilled, composted, burned or fed to animals. At this point, there are no data to calculate the size of this N_2O source.

Methodology for estimating indirect N_2O

Based on the above, we propose the following methodology for calculating a country's indirect N_2O emissions (kg N yr^{-1}):

$$N_2O_{\text{INDIRECT}} = N_2O(G) + N_2O(L) + N_2O(S)$$

where

$N_2O(G) = N_2O$ produced from atmospheric deposition of NO_x and NH_3 (kg N yr^{-1})

$$= (\text{NFERT} * \text{FRACGASF} + \text{NEX} * \text{FRACGASM}) * EF4$$

$N_2O(L) = N_2O$ produced from N leaching and runoff (kg N yr^{-1})

$$= \text{NLEACH} * EF5$$

$N_2O(S) = N_2O$ produced from human sewage (kg N yr^{-1})

$$= \text{NSEWAGE} * EF6$$

$EF4$ = emission factor for atmospheric deposition ($\text{kg } N_2O-N/\text{kg } NH_3-N$ and NO_x-N emitted); Table 2.

$EF5$ = emission factor for leaching/runoff ($\text{kg } N_2O-N/\text{kg N leaching/runoff}$); Table 2.

$EF6$ = emission factor for sewage treatment ($\text{kg } N_2O-N/\text{kg sewage-N produced}$); Table 2.

FRACGASF = fraction of synthetic fertilizer N applied to soils that volatilizes as NH_3 and NO_x ($\text{kg } NH_3-N$ and $NO_x-N/\text{kg of N input}$); Table 3.

FRACGASM = fraction of livestock N excretion that volatilizes as NH_3 and NO_x ($\text{kg } NH_3-N$ and $NO_x-N/\text{kg of N excreted}$); Table 3.

FRACNPR = fraction of N in protein ($\text{kg N/kg of protein}$); Table 3.

FRACLEACH = fraction of N input to soils that is lost through leaching and runoff ($\text{kg N/kg of N applied}$); Table 3.

$$\text{NLEACH} = \text{N leaching in country } (\text{kg N yr}^{-1}) = (\text{NFERT} + \text{NEX} * 0.8) * \text{FRACLEACH}$$

NRPEOPLE = number of people in country. FAO data.

$\text{NSEWAGE} = \text{PROTEIN} * \text{NRPEOPLE} * \text{FRACNPR}$
 PROTEIN = annual per capita protein consumption in country (the global average is $25.6 \text{ kg protein person}^{-1} \text{ yr}^{-1}$).

5. Application of methodology for total N₂O emissions to the world

Total emissions of N₂O from agriculture in 1989 can be calculated as the sum of direct emissions, emissions from animal production and indirect emissions. Thus, Total N₂O–N emissions from the world (kg N₂O–N yr⁻¹) are:

$$\text{N}_2\text{O} = \text{N}_2\text{ODIRECT} + \text{N}_2\text{OANIMALS} + \text{N}_2\text{O INDIRECT}$$

STEP I. Input data collection

Input data:

1. Worldwide use of synthetic fertilizer NFERT = 79 Tg N yr⁻¹ (FAO, 1990b).
2. Number of livestock in world for the following categories: non-dairy cattle, dairy cattle, poultry, sheep, swine and other animals: see FAO data. Total N excretion by livestock NEX = 138 Tg N yr⁻¹ (FAO data; IPCC, 1995b).
3. Dry biomass production of pulses and soybean (CROPBF) assumes 15% water content = 191 Tg x 0.85 = 162 Tg yr⁻¹ (FAO, 1990b).
4. Dry biomass production of other crops (CROP0), assuming 15% water content = 1870 Tg x 0.85 = 1590 Tg/y (FAO, 1990b).
5. Area of organic soils (Histosols) (FOS, ha) = 286 x 10⁵ in tropics, 1459 x 10⁵ in rest of world (Eswaren et al., 1993), assume 10% of Histosols are cultivated in nontropical areas and 10% in tropics.
6. Average annual per capita protein intake in world (PROTEIN) = 25.6 kg person⁻¹ yr⁻¹ (FAO).
7. Number of people in world (NRPEOPLE) = 5.3 billion (FAO).

STEP II: Calculation of fertilizer use (FSN)

Total synthetic fertilizer use excluding emissions of NH₃ and NO_x (FSN):

$$\text{FSN} = \text{NFERT} * (1 - \text{FRACGASF}) = 79 * (1 - 0.1) = 71.1 \text{ Tg N yr}^{-1}.$$

STEP III: Calculation of livestock N excretion (NEX and FAW)

Total animal waste N used as fertilizer (FAW):

$$\text{FAW} = (\text{NEX} * (1 - (\text{FRACFUEL} + \text{FRACGRAZ} + \text{FRACGASM}))) = 138 * (1 - (0.1 + 0.34 + 0.2)) = 49.7 \text{ Tg N yr}^{-1}$$

STEP IV. Calculation of N input from N-fixing crops (FBN)

N fixed by N-fixing crops (FBN):

$$\text{FBN} = 2 * \text{CROPBF} * \text{FRACNCRBF} = 2 * 162 * 0.03 = 9.7 \text{ Tg N yr}^{-1}$$

STEP V. Calculation of N input from crop residues (FCR)

It is assumed that, worldwide, 10% of the crop residue is burned (FRACBURN = 0.1)

Crop residue returned to soils (FCR):

$$\text{FCR} = 2 * [\text{CROP0} * \text{FRACNCR0} + \text{CROPBF} * \text{FRACNCRBF}] * (1 - \text{FRACR}) * (1 - \text{FRACBURN}) = 2 * [1590 * 0.015 + 162 * 0.03] * 0.55 * 0.9 = 28.4 \text{ Tg N yr}^{-1}$$

STEP VI. Calculating direct N₂O emissions

Direct N₂O emissions from agricultural soils:

$$\text{N}_2\text{ODIRECT} (\text{Tg N yr}^{-1}) = [(\text{FSN} + \text{FAW} + \text{FBN} + \text{FCR}) * \text{EF1}] + (\text{Area of tropical cultivated Histisols} * 10 \text{ kg N ha}^{-1} \text{ yr}^{-1}) + (\text{Area of cultivated Histisols in rest of world} * 5 \text{ kg N}_2\text{O-N ha}^{-1} \text{ yr}^{-1}) = [(71.1 + 49.7 + 9.7 + 28.4) * 0.0125] + [0.03 + 0.07] = 2.0 + 0.1 = 2.1 \text{ Tg N yr}^{-1}.$$

STEP VII Calculating N₂O emissions from animal production

N₂O emissions from animal production (N₂OANIMALS in Tg N yr⁻¹) as:

$$\text{N}_2\text{O ANIMALS} = \text{N}_2\text{O(T,AWMS)} = [\text{N(T=1)} * \text{NEX(T=1)} * \text{AWMS(T=1)} * \text{EF3(AWMS)}] + \dots + [\text{N(T=TMAX)} * \text{NEX(T=TMAX)} * \text{AWMS(T=TMAX)} * \text{EF3(AWMS)}] = 2.1 \text{ Tg N yr}^{-1}.$$

STEP VIII. Calculating indirect emissions

Indirect N₂O emissions (kg yr⁻¹) can now be calculated (1) (atmospheric deposition), (2) (leaching and runoff) and (3) (human sewage) as: N₂OINDIRECT = N₂O(G) + N₂O(L) + N₂O(S) = [(NFERT*FRACGASF+NEX*FRACGASM)*EF4] + [(NFERT + NEX *FRACLEACH) * EF5] + [(PROTEIN*NRPEOPLE*FRACNPR) * EF6] = [0.3 + 1.6 + 0.2] = 2.1 Tg N yr⁻¹

STEP IX. Total N₂O emissions from agriculture

Total emissions of N₂O from agriculture can be calculated as the sum of direct emissions, emissions from

Table 7. Global N₂O emissions from agricultural soils calculated with the PHASE II methodology (Tg N yr⁻¹)

Direct soil emissions	
- synthetic fertilizer	0.9(0.18–1.6)*
- animal waste	0.6(0.12–1.1)
- biological N ₂ fixation	0.1(0.02–0.2)
- crop residue	0.4(0.07–0.7)
- cultivated Histosols	0.1(0.02–0.2)
- total	2.1(0.4–3.8)
Animal production	
- animal waste management systems	2.1(0.6–3.1)
Indirect emissions	
- atmospheric deposition	0.3(0.06–0.6)
- nitrogen leaching and runoff	1.6(0.13–7.7)
- human sewage	0.2(0.04–2.6)
- total	2.1(0.23–11.9)
Total	6.3(1.2–17.9)

^a Values in parentheses indicate estimated range which is derived from emission factor range in Table 2.

animal production and indirect emissions. Total N₂O–N emissions are calculated to be 6.3 Tg N yr⁻¹ (Table 7).

6. Future needs for methodology development

The methodology for country-based N₂O described above is a rough, generalized approach which treats all agricultural systems as being the same under all climates, in all soils, in all crops and in all management systems. This clearly provides uncertainties in inventory calculations. We do feel, however, that the ranges of conversion factors provided cover much of the potential N₂O emissions from each country, whatever climate, soils and set of crops is involved. Some recent studies in temperate (e.g. Thornton and Valente, 1996) and tropical (Veldkamp and Keller, 1997) show very high direct N₂O emissions while other studies (Corrie et al., 1996; Flessa et al., 1995; Wagner-Riddle and Thurtel, this issue) demonstrate that significant N₂O emissions commonly occur during thaw periods in early spring and winter or through snow-covered agricultural soils (Van Bochove et al., 1996). These recent studies indicate that many of the published direct estimates of N₂O emissions from agricultural fields may be further underestimated because they did not account for cold season N₂O emissions, which can be substantial.

To make significant improvement in inventory methodologies for N₂O, we felt that the next step is

to utilize process based models to produce country inventories, for direct emissions from agricultural soils (e.g. Li et al., 1992; Potter et al., 1996; Parton et al., 1996), appropriate animal management models for N₂O from animal production, simulation models which more effectively represent N transformations in aquatic systems, including riparian areas, wetlands, rivers estuaries, continental shelves and the deep ocean (e.g. Seitzinger and Kroeze, submitted).

The soil C and N cycles are tightly integrated and we think that both C and N should be considered together so that various aspects of the C and N cycle and CO₂ and N₂O production can be more accurately defined. For example, the amount of N leached from agricultural fields represents a very large component of the global N₂O production according to the methodology presented here. The accuracy of the N leaching fraction prediction is closely tied to C turnover in the soil as it controls N mineralization and immobilization. The turnover and retention of N in all soils is intimately linked with the C cycle. Conversely, C retention in soils is directly tied to mineral N availability. These models must, however, include adequate flexibility to predict cold soil emissions as well as emissions under tropical conditions.

Testing of the validity and accuracy of any inventory methodology needs to be conducted. Van Aardenne (1996) conducted an uncertainty and sensitivity analysis on the methodology presented in this paper. Using a Monte Carlo sampling technique and regression and correlation analyses he simulated the output of the methodology and determined the total uncertainty of the simulated model output and uncertainty contribution of 14 model parameters used for calculating emissions from soils, grazing animals, and indirect emissions. He found a standard deviation of 20% from calculated mean N₂O emission values using the 1990 Dutch N values as a test situation. He discerned that the parameters EF1 (emission factor for direct soil emissions) and the FLH (expression of the fraction of N input to soils that is lost through leaching and runoff) were the largest contributors to the overall methodology uncertainty. This rather low estimate of error for the methodology is surprisingly small and suggests the need for further error analyses using country data that include a variety of climate and agricultural systems.

There are additional issues that should be addressed in the overall effort of estimating greenhouse gas exchange with agricultural systems. These issues include: (1) development of methodologies that

represent the effect of cropping system, soil, and climate on N_2O budgets which account for N_2O emissions per unit of food produced in various agricultural systems; (2) including soil methane oxidation in national budgets (without the soil sink component atmospheric methane concentrations would be increasing about two times faster than the increase rate observed in the 1980s) (Ojima et al., 1993); (3) including the impact of NO_x emissions from agricultural soils on local and regional atmospheric oxidants and ozone concentration; (4) determining the impact of C and N losses and retention on system sustainability; (5) considering mitigation methodologies to decrease CO_2 and N_2O emissions from agriculture and to improve the soil sink capacity for CH_4 ; and (6) investigating errors that may arise as a result of aggregating field scale data to the national level. We assume that the methods used are applicable to small and large countries. However, there can be effects of 'scaling' for which there is no quantification.

7. Summary

This paper presents the development of a new method for calculating national emissions of N_2O from agricultural soils. This N_2O method is a revision of the 1995 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 1995b). The N_2O methodology was developed a group of experts, in which approximately 32 people from 18 countries participated.

The methodology includes more sources of N_2O and more explicit information about emission factors to be used than the Phase I IPCC Guidelines (IPCC, 1995). Three sources of N_2O are distinguished: (i) direct emissions from agricultural soils, (ii) emissions from animal production systems, and (iii) N_2O emissions indirectly induced by agricultural activities. The methodology is a simple approach which requires only input data that are available from FAO databases.

Our methodology attempts to relate N_2O emissions to the agricultural nitrogen cycle and to systems into which N is transported once it leaves agricultural systems (Figure 1). Anthropogenic input into agricultural systems include synthetic fertilizer, N from animal wastes, N from increased biological N-fixation, N from crop residue returned to the field and N derived from cultivation of mineral and organic soils through enhanced organic matter mineralization. Nitrous oxide may be produced and emitted directly in agricultural fields, animal confinements or pastoral

systems. Part of the N applied to agricultural soils or from animal production systems is transported from agricultural systems into ground and surface waters through surface runoff, N leaching, consumption by humans and introduction into sewage systems which transport the N ultimately into surface water. Additional N_2O is produced during transport and residence in aquatic systems. Ammonia and oxides of N (NO_x) are also emitted from agricultural systems and may be transported off-site and serve to fertilize other systems which leads to enhanced production of N_2O .

The methodology has a number of limitations because agricultural systems are considered as being the same throughout the world, and the methodology does not take into account different crops, soils and climates which are known to regulate N_2O production. It is likely that direct field emissions during the winter time from temperate soils as well as some tropical situations are underestimated. These factors are not considered because limited data are available to provide appropriate emission factors. The method also uses a linear extrapolation between N_2O emissions and fertilizer N application and in the indirect emissions section does not account for the probable lag time between N input and ultimate production of N_2O as a result of this N input into agricultural soils.

We estimated global N_2O -N emissions for the year 1989, using midpoint emission factors from our methodology and the FAO data (FAO, 1990a,b) (Table 7). Direct emissions from agricultural soils totaled 2.1 Tg N, direct emissions from animal production totaled 2.1 Tg N and indirect emissions resulting from agricultural N input into the atmosphere and aquatic systems totaled 2.1 Tg N_2O -N. These estimates show that each of the three components of agriculture considered contribute about the same amount of N_2O to the global atmospheric budget. When looking at previous estimates (Table 1), the N_2O input to the atmosphere from agricultural production as a whole has apparently been previously underestimated. We found that the total global N_2O budget is reasonably in balance if we use the N_2O emission estimate for agricultural soils calculated by the methodology presented in this paper (Table 1). The balance is tenuous, however, because of the large uncertainty in our estimate of global N_2O from agriculture (Table 7) and the proportionation of natural and anthropogenic sources which needs to be reconciled. There are a few potential overlaps in sections of our estimates and those from IPCC 1994. For example, we estimate N_2O emissions from agricultural N that flows from rivers into oceans.

Part of the 'anthropogenic' N_2O is termed as 'natural' oceanic N_2O in the IPCC 1994 estimates. Also, because of the emissions of ammonia and nitrogen oxides from agricultural sources, and their eventual redeposition onto 'natural' systems, N_2O emissions from natural ecosystems may include N_2O derived from N evolved from agricultural soils and redeposited onto forest and grasslands. Exact quantification for these overlaps is difficult, and may have resulted in double counting of some sources. These new estimates suggest that the missing N_2O source discussed in earlier IPCC reports (Table 1) is likely to be a biogenic (agricultural) one.

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Appendix 1. Default Values for Percentage of Manure N Produced in Different Animal Waste Management Systems in Different World Regions (from Safley et. al., 1992)

Region	Type of Animal	Percent of Manure Production per Animal Waste Management Systems						
		Anaerobic Lagoon	Liquid System	Daily Spread	Solid Storage and Drylot	Pasture Range and Paddock	Used Fuel	Other System
North America	Non-dairy Cattle (D)	0	1	0	14	84	0	1
	Dairy Cattle	10	23	37	23	0	0	7
	Poultry (E)	5	4	0	0	1	0	90
	Sheep	0	0	0	2	88	0	10
	Swine	25	50	0	18	0	0	6
	Other animals (F)	0	0	0	0	92	0	8
Western Europe	Non-dairy Cattle (D)	0	55	0	2	33	0	9
	Dairy Cattle	0	46	24	21	8	0	1
	Poultry (E)	0	13	0	1	2	0	84
	Sheep	0	0	0	2	87	0	11
	Swine	0	77	0	23	0	0	0
	Other animals (F)	0	0	0	0	96	0	4
Eastern Europe	Non-dairy Cattle (D)	8	39	0	52	0	0	1
	Dairy Cattle	0	18	1	67	13	0	0
	Poultry (E)	0	28	0	0	1	0	71
	Sheep	0	0	0	0	73	0	27
	Swine	0	29	0	0	27	0	45
	Other animals (F)	0	0	0	0	92	0	8
Oceania	Non-dairy Cattle (D)	0	0	0	0	100	0	0
	Dairy Cattle	0	0	0	0	100	0	0
	Poultry (E)	0	0	0	0	3	0	98
	Sheep	0	0	0	0	100	0	0
	Swine	55	0	0	17	0	0	28
	Other animals (F)	0	0	0	0	100	0	0
Latin America	Non-dairy Cattle (D)	0	0	0	0	99	0	1
	Dairy Cattle	0	1	62	1	36	0	0
	Poultry (E)	0	9	0	0	42	0	49
	Sheep	0	0	0	0	100	0	0
	Swine	0	8	2	51	0	0	40
	Other animals (F)	0	0	0	0	99	0	1
Africa	Non-dairy Cattle (D)	0	0	1	3	96	0	0
	Dairy Cattle	0	0	12	0	83	0	5
	Poultry (E)	0	0	0	0	81	0	19
	Sheep	0	0	0	1	99	0	1
	Swine	0	7	0	93	0	0	0
	Other animals (F)	1	0	0	0	99	0	1
Near East and Mediterranean	Non-dairy Cattle (D)	0	0	2	0	77	18	2
	Dairy Cattle	0	0	3	3	77	18	0
	Poultry (E)	0	1	0	0	71	0	28
	Sheep	0	0	0	0	100	0	0
	Swine	0	32	0	68	0	0	0
	Other animals (F)	0	0	0	0	100	0	0

Appendix 1. Continued

Region	Type of Animal	Percent of Manure Production per Animal Waste Management Systems						
		Anaerobic Lagoon	Liquid System	Daily Spread	Solid Storage and Drylot	Pasture Range and Paddock	Used Fuel	Other System
Asia and Far East	Non-dairy Cattle (D)	0	0	16	14	29	40	0
	Dairy Cattle	6	4	21	0	24	46	0
	Poultry (E)	1	2	0	0	44	1	52
	Sheep	0	0	0	0	83	0	17
	Swine	1	38	1	53	0	7	0
	Other animals (F)	0	0	0	0	95	0	5

(D) Includes buffalo.

(E) Includes chickens, turkeys and ducks.

(F) Includes goats, horses, mules, donkeys and camels.